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Population Risks from Uranium Ore Bodies

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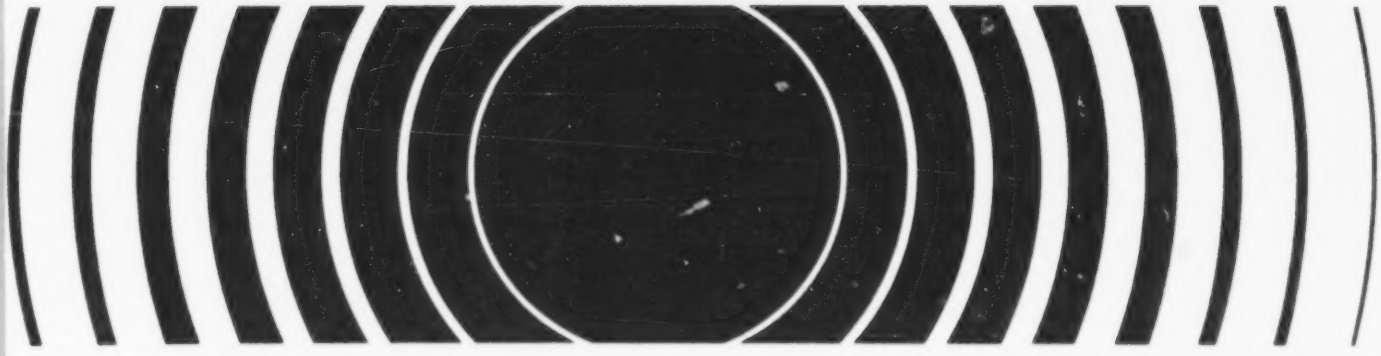

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High-Level Waste Environmental
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Population Risks from Uranium Ore Bodies

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October 1980

Office of Radiation Programs
U.S. Environmental Protection Agency
Washington, D.C. 20460

PREFACE

The Office of Radiation Programs, U.S. Environmental Protection Agency, carries out a national program to evaluate individual and population exposure to ionizing and nonionizing radiation and to promote development of controls for the protection of public health and the environment.

This report is technical support for EPA's high-level radioactive waste standards program; it estimates the radiological releases and impact of unmined uranium ore.

The Office of Radiation Programs invites readers to report omissions or errors, submit comments, or request further information.

OFFICE OF RADIATION PROGRAMS
U.S. ENVIRONMENTAL PROTECTION AGENCY

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ABSTRACT

This report estimates the minimum radiological releases and potential impact of deep-lying uranium ore, so that they may be compared with projected releases and impacts from radioactive waste. Uranium concentrations and groundwater flow rates are used as input data for three models developed by EPA for analyzing the impact of high-level radioactive waste. One set of data is obtained from some ore bodies which are being mined by the in situ solution process. Another, the minimum impact case, is obtained by using conservative data on uranium concentrations in uraniferous groundwaters in conjunction with a model aquifer developed by EPA.

The estimated releases from actual ore bodies prior to mining range from .19 to 1.1 curies per year for ^{226}Ra. For a generic model ore body with parameters chosen to minimize the impact, a release of 1.2×10^{-4} curies ^{226}Ra per year is estimated. To take into account the different size of these ore deposits, the impacts can be adjusted or normalized to the amount of ore from which a 100,000-metric-ton heavy metal nuclear waste repository would be derived. After normalization, the estimated yearly releases range from 7.4 millicuries for the minimum case to 30 or 300 curies for the actual ore bodies. This corresponds to .023 fatal cancers per year for the minimum case and 100 to 1000 fatal cancers per year for the actual ore bodies using EPA environmental pathway and dosimetry models.

The approach used in this work can serve as a starting point for making more detailed assessments using other assumptions.

CONTENTS

	Page
Preface	iii
Abstract	v
Introduction	1
Method	3
Environmental Impact Analysis--Basic Assumptions	8
Generic Model of Natural Ore Body	10
Site-Specific Case Studies	14
Discussion	19
Conclusion	22
References	24

Tables

I.	Uranium, Daughters, and their Retardation and Dose Conversion Factors	5
II.	Data Used for Individual Case Studies	12
III.	Estimated and Normalized Health Effects for Site Specific and Generic Uranium Ore Bodies	15

Introduction

Many authors have compared the health hazard from unmined uranium ore with that from nuclear waste. Several (Cohen, 1977; Hamstra, 1975; and the review by Voss, 1979) believe that the hazard of high-level radioactive waste after decay of several hundred years is less than that of unmined uranium ore. Their comparisons used hazard indices which do not indicate how these materials behave in the natural environment.

This assessment estimates the releases and impact of actual, natural uranium ore bodies, rather than just a measurement of potential impact. For this purpose, I have used an assessment methodology incorporating models that EPA is using to assess the impacts of high-level waste repositories. In addition to examining a few actual ore bodies, this report presents an approach for estimating the minimum radiological impact from unmined uranium ore. The report deals only with the population risk from unmined ore bodies and not with the risks and benefits from the nuclear generation of electricity or from other uses of nuclear energy.

Most of the uranium ore bodies in the contiguous portion of the United States are secondary deposits of waterborne uranium in

permeable geologic formations. The source of the uranium was igneous, uranium-bearing rocks, which were oxidized or weathered and later leached by water. The dissolved uranium moved with the water. In some cases the waterborne uranium reached the ocean; in others the solution infiltrated the ground and either the uranium was deposited in permeable rock that was strongly reducing or the solution was mixed with agents (such as phosphates or vanadates) forming insoluble uranium salts. The factors that control the deposition of uranium are diverse but interrelated: the oxidation-reduction potential (Eh), acidity (pH), presence of complexing and/or precipitation agents, and the presence of adsorbing solids, such as marine muds, iron oxyhydroxides, clays, and organics. Despite this diversity of mechanisms of uranium deposition, almost all U.S. uranium reserves currently commercial at production costs not exceeding $50 per pound are found in coarse clastic rocks (DOE, 1979), where uranium was deposited from ground water. Qidwai and Jensen (1979) and Deffeyes and MacGregor (1980) discuss the formation of these "roll front" sandstone deposits in further detail.

After deposition of uranium, the daughter products will accumulate according to the laws of radioactive decay. If the ore deposit is in a permeable geological medium, it is possible for both

the uranium and its daughters to dissolve into water moving through the deposit (See, for example, Fix, 1955; Phoenix, 1959; Germanov and Panteleyev, 1968; Kaufmann et al., 1975; Langmuir, 1978; Capuano, 1978; Lueck, 1978; or Runnells et al., 1980). This dissolution can also be selective; activity ratios of $^{234}U/^{238}U$ in groundwater as high as 12 have been reported (Fleisher and Raabe, 1978).

The remainder of this report presents a methodology for estimating the radionuclide releases from an ore body, the transport of the radionuclides, and the resulting effect on people. The methodology is applied to a model ore body designed to estimate a lower bound of its effects and also to actual ore bodies to get a more realistic, but probably high, effect estimate.

Method

The method used in this paper consists of applying three models. One considers the transport of radionuclides from an ore body to a stream via groundwater. Another considers the movement of radionuclides in the biosphere once released to a stream. The third estimates the effect of released radionuclides on people. Uranium concentrations and groundwater flow rates can be used as input data to these three models.

The geosphere transport, biosphere transport, and dosimetric models (Smith et al., 1980a and Smith et al., 1980b) were developed initally to analyze the consequences of high-level nuclear waste burial. The biosphere transport and dosimetric models consider various means of contamination reaching people. These include direct exposure, drinking water, eating fish, eating irrigated food (crops, beef, milk), and breathing air. Table I summarizes the results of Smith et al. (1980b), who estimated the number of fatal cancers that would result in their model population for each curie of various radionuclides entering a stream. They gave 3.1 cancers/curie of ^{226}Ra and 1.3 cancers/curie of ^{234}U. In this analysis I consider only the impact of ^{226}Ra. Since ^{226}Ra is reconcentrated in groundwater (as discussed later) its effect will be larger than that of any other daughter of ^{238}U. This simplification will not affect a minimum impact estimate, since the contribution of other uranium daughters can only increase the radiological impact. Radon-222 (the most significant air pollutant in uranium ore) has a half-life of 3.8 days. Radon in groundwater would probably decay before reaching the surface. Since all nuclides except ^{226}Ra are ignored, my estimate of health effect will be low.

One set of data used in this modeling is taken from several ore bodies being exploited by the in situ (solution mining) process.

Table I

Uranium, Daughters, and their Retardation and Dose Conversion Factors

Isotope	Half-life	Retardation Factor[a,c]	Fatal Cancers/Ci Release[b] to Stream
^{238}U	4.5×10^{9}y	14,300	-
^{234}U	2.5×10^{5}y	14,300	1.3
^{230}Th	7.8×10^{4}y	50,000	-
^{226}Ra	1.6×10^{3}y	500	3.1
^{222}Rn	3.8 days	1	-

[a]Source: Arthur D. Little (1979); originally from Denham et al. (1973).

[b]Source: Smith et al. (1980a, 1980b).

[c]Retardation factor:

$$\frac{\text{Time in years a given radionuclide takes to travel 10 feet}}{\text{Time in years groundwater takes to travel 10 feet}}$$

The amount of uranium leached from an ore body per year before mining began is the product of the annual volume of water moving through it times the measured premining concentration of uranium in the groundwater. Another set of inputs is obtained by using lower bound data for uranium concentrations in uraniferous groundwaters (Fix, 1955; Germanov and Panteleyev, 1968; Langmuir, 1978; Capuano, 1978) in conjunction with EPA's model aquifer, which is one mile long (Smith et al., 1980a).

The total annual amount of water moving through an ore body is given by the velocity of the water times the porosity of the ore body times the cross-sectional area. The quantity of uranium leached by the groundwater will be the product of the amount of water and the concentration of uranium in the water. All of this data is routinely measured for economic deposits.

The dissolved uranium and daughters will move down the hydrologic gradient slower than the groundwater because they react chemically and physically with the rock. Retardation factors in a desert soil are given in Table I for ^{238}U and some of its daughters. If one assumes a groundwater velocity of 10 meters per year, ^{230}Th will travel 10 meters in 50,000 years; ^{226}Ra will move 10 meters in 500 years; and ^{222}Rn, because of its short half-life, will decay before it moves 10 meters. Since it takes a significant fraction of their half-lives for the ^{230}Th and the ^{226}Ra to move this short distance, they will decay before they are transmitted long distances in the groundwater.

Uranium-238 and its daughters are present in all the aquifer since uranium's long half-life permits it to move long distances from the source ore body. Radium-226 is found at higher activity concentrations in the groundwater than ^{238}U because of differential

adsorptions of elements by the rocks (Burkholder and Cloninger, 1977, 1978; Lester et al., 1975). Approximating the decay chain by $^{238}U \rightarrow ^{234}U \rightarrow ^{230}Th \rightarrow ^{226}Ra$, and using the approximation method outlined by Rogers (1978), the ratio between the ^{226}Ra activity and the initial ^{238}U activity released will be less than or equal to 39 when the retardation factors are those in Table I, the travel time of the groundwater is 290 years (the shortest time to travel one mile of any reported for the specific cases later examined), and the preferential enrichment ratio of ^{234}U to ^{238}U is 12 (the largest reported by Fleisher and Raabe (1978)). The exact solution for the four-member decay chain shown above is very difficult. However, the solution of a three-member chain has been solved by Lester et al. (1975) and has been applied to this system:

Uranium-238 → Thorium-230 → Radium-226

The enrichment factor of ^{226}Ra relative to ^{238}U is highly dependent on the retardation factors, aquifer velocity, and travel distance. The solution of Lester et al. (ignoring dispersion) shows an activity for ^{226}Ra that is 33 times higher than the initially released ^{238}U activity using the data shown above. The activity enrichment factor will be used for some cases but may overestimate the enrichment. Since this enrichment factor changes when the

retardation factors, distance, or travel time change, an enrichment factor of 10 will be used for the minimum case, to make certain this case is conservatively low.

Environmental Impact Analysis--Basic Assumptions

To analyze the impact, one must know the boundary conditions at the ore body and the geochemical characteristics of the aquifer. It is difficult to determine these characteristics precisely. I assume that (1) solubility considerations limit the uranium concentration in groundwater at one part per billion, (2) uranium leaves the ore body at a rate of solubility times water flow, (3) all of the uranium released from the ore body reaches the stream at the same rate it enters groundwater, (4) ^{226}Ra reaches the stream at activity levels either 10 or 33 times higher than the ^{238}U activity, and (5) one curie of ^{226}Ra released to the stream causes a commitment of 3.1 fatal cancers.

This model assumes that solubility controls the concentration of aqueous uranium in the ore at one part per billion. The concentration of uranium in the groundwater of ore bodies has been shown to decrease down dip, after water has moved through the ore body (Germanov and Panteleyev, 1968 and Phoenix, 1959). This argues

strongly that thermodynamics are more important in limiting the uranium concentration rather than the kinetics discussed by Grandstaff (1976) at higher concentrations. For any given daughter of ^{238}U it is reasonable to assume that after several million years, the daughter will be in equilibrium with the parent as they both migrate in an aquifer, except that the daughter might migrate faster or slower as discussed above. Groundwater chemistry can change radically along an aquifer and this can change the amount of dissolved uranium. For example, a rise in the solution CO_2 pressure from $3x10^{-4}$ atmospheres to 10^{-2} atmospheres increases the solubility of uraninite (for Eh values above -.05 Volts) by over 1000 times (Langmuir, 1978). Despite the uncertainty of the data, there is evidence that the groundwater in some ore bodies is saturated or even supersaturated with some uranium mineral species (Boberg and Runnells, 1971; Lueck, 1978; Runnells et al., 1980).

If one assumes that uranium is dissolved by groundwater at a known rate per year and travels 1 mile to a surface stream, the annual release rate to the stream will eventually be the same as the release rate from the ore body. The release rate of ^{238}U will control the release rate of the daughters to the stream, because the daughters dissolved directly from the ore body do not migrate fast enough to reach the stream prior to decay. Although a travel

distance of 1 mile has been used throughout this work, the long half-life of ^{238}U obviously permits it to be transmitted for much longer distances in groundwater.

In order to compare the impact of ore bodies with that of a high-level waste repository, the two must be put on a comparable basis. The amount of fuel used by a 1000 MWe reactor in 1 year is about 35 metric tons enriched uranium; therefore, a 100,000-metric-ton heavy metal repository will contain waste equivalent to 2900 reactor years of operation. Because 1140 metric tons unenriched U_3O_8 are the annual requirements for 5.3 1000 MWe reactors (EPA, 1973), the annual requirement per reactor is 215 MT unenriched U_3O_8. Therefore, a 100,000-metric-ton heavy metal repository inventory was derived from 620,000 metric tons U_3O_8 (equal to 215 metric tons per reactor year times 2900 reactor years).

Generic Model of Natural Ore Body

I propose a generic model to estimate the minimum impact of an unmined uranium ore body. The data used for this case is summarized in Table II.

The model ore body is located in an aquifer that has a slower velocity than those of the actual ore bodies considered later. The

model aquifer is the upper, or overlying, aquifer in the model high-level waste repository developed by EPA (Smith et al., 1980a and Arthur D. Little, Inc., 1979). The aquifer has the following characteristics:

horizontal permeability = 10^{-4} cm/sec,
gradient = .01,
porosity = 15%, and
thickness of strata = 30 meters

The cross-sectional width of the model ore body is 3.7 kilometers, perpendicular to the groundwater flow. This width has been selected because 1) it falls within the range of actual ore bodies listed in Table II, and 2) it is the approximate width of a high-level waste repository (as described by Smith et al., 1980a and Egan, 1978). One would expect a roll front deposit to form perpendicular to the flow of groundwater. The interstitial velocity in this generic ore body will approximate 2.1 meters/year. Since the width of the ore body is 3.7 kilometers, the annual flow of water through the hypothetical ore body will be 35,000 cubic meters. If this entire volume of water leaves the ore body with a concentration of one part per billion of uranium, the amount of uranium leached per year will be 35 grams/year (12 microcuries/year of ^{238}U). A

Table II

Data Used for Individual Case Studies

Site	Interstitial Groundwater Velocity (meters/year)	Thickness of Ore Strata (meters)	Cross Sectional Width of Ore Body (meters)	Porosity	Uranium Concentration (ppm)	Uranium Released (mCi/y)	Estimated Uranium Reserves (metric tons U_3O_8)
Model Case	2.1	30	3,700	.15	.001	.012	10,000
Nine Mile Lake Natrona Co., Wyo.	5.3	20	5,600	.28	[a].1	5.7	2,000
Irigaray Johnson Co., Wyo.	4.3	37	11,000	.23	.1	13	5,900 - 6,800
Highland (Case 1) Converse Co., Wyo.	5.5	40	[b]1,500	.30	.2	6.7	450 - 1,400
Highland (Case 2) Converse Co., Wyo.	5.5	40	[c]7,600	.30	.2	33	25,000

[a]Measured values range from .640 to levels not detectable.

[b]Assumes all groundwater flow is to the southeast and ignores the effects from dewatering for mining operations in the area. This includes ore reserves only from the solution mining operations.

[c]This includes ore and reserves from the entire Exxon Highland operation and ore and reserves from the adjacent Morton Ranch project and assumes that all water flow is to the southeast. This may not be true because mine dewatering has changed the site hydrology. Reserves are as reported in the environmental reports of the projects.

uranium concentration of 1 part per billion is lower than that measured in uranium mines and in uraniferous aquifers (Phoenix, 1959; Langmuir, 1978; Germanov and Panteleyev 1968; Kaufmann et al., 1975; Runnells et al., 1980) and is the lowest value reported for uraniferous aquifers by Fix (1955). It compares favorably with the thermodynamic values of Capuano (1978), and Langmuir (1978).

It might be argued that the location of the model ore body in an aquifer is inappropriate since the presence of the aquifer would tend to maximize the impact by providing a ready source of geological transport. However, over 90 percent of known U.S. uranium reserves (at production costs not exceeding $50 per pound) are found in coarse clastic sediments (DOE, 1979), terrains in which aquifers would be expected.

Additionally, it is possible that ^{226}Ra can be reconcentrated into aquifers while it and its parents migrate, as discussed previously. To make certain that the model produces a lower bound, an activity enrichment factor of 10 will be used for this case, rather than the 33 obtained from the decay and diffusion equations discussed earlier. Using this factor, the projected yearly release of ^{226}Ra to the stream is 120 microcuries, which produces a commitment of 3.7×10^{-4} fatal cancers per year (1.2×10^{-4}Ci x 3.1

cancers/Ci) (Smith et al., 1980a and Smith et al., 1980b). If other radionuclides in the decay chain were considered, the impact would be larger, but this fact does not affect this minimum estimate. The release and impact of the model ore body are summarized in Table III.

For the sake of comparison with the actual ore bodies discussed subsequently, the model ore body is assumed to have reserves of 10,000 metric tons U_3O_8. This corresponds to an ore body 50 meters thick, 30 meters high, and 3.7 kilometers wide, with an ore grade of .09% (assuming the density of the host is 2g/cc). This ore grade compares favorably to the average of .07% for ore reserves (at costs not exceeding $50 per pound) reported by DOE (1979). If the ore reserves were smaller, the impact per ton of U_3O_8 would be larger after normalization. This normalization process puts the effects of different size ore bodies on a comparable basis. This adjustment is done by dividing the effects of an ore body by its reserves and multiplying the quotient by 620,000 metric tons, the amount of uranium from which a 100,000-metric-ton heavy metal repository was derived.

Site-Specific Case Studies

The Nine Mile Lake, Irigary, and Highland sites have been selected for site-specific analysis, since field data are available

Table III

Estimated and Normalized Health Effects for Site Specific and Generic Uranium Ore Bodies

Site	Uranium Released (mCi/y)	Radium Released (mCi/y)	Estimated Fatal Cancers from Radium Released (per year)	Normalized Effects[a]
Generic Model Ore Body (Minimum case)	.012	.12	3.7x10-4	2.3×10^{-2}
Nine Mile Lake Natrona Co., Wyo.	5.7	190	.59	180
Irigaray Johnson Co., Wyo.	13	430	1.3	120-140
Highland (Case 1) Converse Co., Wyo.	6.7	220	.68	300-940
Highland (Case 2) Converse Co., Wyo.	33	1100	3.4	84

[a]Estimated yearly fatal health effects for ore from which a 100,000 MTHM radioactive waste repository was derived.

from them. The adsorption of the radionuclide by the geological medium plays an important role in retarding the migration of radionuclides. The results of laboratory tests on the retardation factors indicate that this parameter may vary greatly because of the

chemical characteristics of the solution and the geological and geochemical characteristics of the geological medium. The retardation factors used are shown in Table I. These are for a typical western desert soil with low-to-moderate cation exchange capacity. This sandy-to-sandy-loam soil contained about one milligram of free $CaCO_3$ per gram (Denham et al., 1973).

The site characteristics for the specific cases studied are summarized in Table II. I used premining hydrologic and chemical data obtained from some ore bodies now being mined by the in situ solution method. These data are probably superior to that from other kinds of uranium mines. In Table III the cancer commitment is summarized in terms of the ores from which a 100,000 metric-ton-heavy metal repository was derived. The cancers were obtained by multiplying the ^{226}Ra releases in curies/year by 3.1 cancers/curie (Smith et al., 1980a, and Smith et al., 1980b). The contribution from all other isotopes is relatively insignificant because of the large enrichment of ^{226}Ra relative to ^{238}U in the groundwater. Since only premining data has been used in this work, it does not assess the environmental impact of current mining operations. These actual ore bodies are in arid regions where the population density is lower than the population density in the EPA model.

Nine Mile Lake:

Using premining data from the Rocky Mountain Energy Company's Nine Mile Lake *in situ* solution-mining operation, the ore body released about 190 millicuries of ^{226}Ra per year before mining began. This release causes .59 fatal cancers per year using EPA's model pathways and populations. The reserves shown in Table II will permit operation of the proposed 227 metric tons per year plant for 9 years (Rocky Mountain Energy Co., 1979).

Irigaray Project:

Using data from Wyoming Mineral Company's Irigaray *in situ* solution-mining project, an annual release of 430 millicuries of ^{226}Ra per year occurred in the ore body prior to mining. An average of 1.3 fatal cancers per year might be expected from this release using EPA's model pathways and populations (Wyoming Minerals Corp., 1978).

Highland Project:

The Highland Project of Exxon Minerals is divided into two cases. In one of these only the release from the ore bodies currently being mined by the *in situ* method is considered (220

millicuries of ^{226}Ra per year); in the other the release of all the ore bodies in both the Highland and adjacent Morton Ranch projects are assessed (1100 millicuries of ^{226}Ra per year). These releases would cause a yearly commitment of .68 and 3.4 fatal cancers. The site specific chemical and hydrologic data developed for the original Highland Mill have been supplemented with more recent data from the solution-mining proposal. Mine dewatering operations have changed the local hydrology of the area. Because of this uncertainty, the first available report on hydrology has been used, although later references show very different measurements (see, for example, Final Environmental Statement for Highlands Uranium Solution Mining Project, pages 2-14 and 2-16, Exxon Minerals Co., 1978).

The stated uranium reserves for the second case are probably too low because the price of uranium has increased substantially since the Highland mill was licensed. Of the ore bodies at the Highland area, 1 to 3 million pounds U_3O_8 are reported as the Highland solution-mining reserves, 32 million pounds U_3O_8 are from the Highland surface operations, and 21 million pounds are attributed to the adjacent Morton Ranch mines. These reserves total about 55 million pounds or 25 thousand metric tons.

Discussion

Measured concentrations in uraniferous areas are generally higher than the assumed concentration (one part per billion) of uranium in the model. This is the same concentration used in some risk analysis cases for a model high-level radioactive waste repository (Smith et al., 1980a); increases or decreases in this solubility-limited concentration would affect uranium in both models in a parallel manner. Although the concentration of uranium in a saturated solution can be lower under some conditions (Langmuir, 1978), field measurements and some thermodynamic studies show this one-part-per-billion limit to be a reasonable uranium concentration in the reducing down-dip groundwater (Kaufmann et al., 1975; Fix, 1955; Phoenix, 1959; Germanov and Panteleyev, 1968; Capuano, 1978). Additionally, factors which might vary, such as changes in cancers per rem, retardation factors, or target populations, would affect both the repository and ore body impacts in a parallel, although not necessarily identical, manner.

The estimated effects from the actual ore bodies considered have great uncertainty. This is because (1) trace concentrations of uranium are difficult to measure and vary within the ore, (2) small changes in Eh can produce large changes in dissolved uranium

concentrations, (3) the groundwater transport and dosimetric models are, in part, hypothetical (Smith et al., 1980a) and (4) the retardation factors used are from one medium and might not reflect the range of values that exist in nature. The area of largest potential uncertainty is in interpreting the available data for uranium solubility, which can vary by several orders of magnitude.

The release and impact assessments for the actual ore bodies should be looked upon as high estimates for the groundwater pathway because the uranium concentrations used are higher than one might expect down dip. Although the actual impact of these ore bodies could be higher, due to limitations of the transport and dosimetric models, it is more likely that the impact would be lower, because uranium solubility decreases as the groundwater migrates down dip into a more reducing geological environment (Phoenix, 1959 and Germanov and Panteleyev, 1968). Since the reported uranium concentrations are averages, they may include samples taken from both the oxidizing and reducing side of the ore body. It is my judgment that the averages reported for the actual ore bodies are overestimates for this reason. Additionally, it is possible that drilling in the ore body, to obtain samples, introduced oxygen, causing an increase in the uranium concentration. Therefore, these estimates can be looked upon only as a high impact estimate, since

the high uranium concentration tends to maximize the estimated releases and impacts.

The releases from unmined uranium ore bodies to groundwater are "chronic" or continuous. If one postulated or assumed various accidents (such as drilling into the ore), the impact might be much larger, if no environmental or institutional controls are assumed.

The impact estimate for the actual ore bodies is over a thousand times higher than that from the model ore body. The actual effects from deep ore bodies would probably fall between these estimates because the uranium concentrations reported from the actual ore bodies are not typical of the down-dip concentrations. If the uranium concentration is raised by a factor of 120 (the highest reported by Fix (1955)) and the reconcentration factor is raised by a factor of 3.3, the estimated effects from the model ore body fall near the range found for the actual ore bodies.

The differences in the impacts can be explained: a) there are factors which make the effects from the model ore body smaller: solubility of uraninite (2 orders of magnitude), interstitial aquifer velocity (factor of 2 more), enrichment of radium-226 in aquifer relative to parents (factor of 3), and cross-sectional width

of the ore body (factor of 2); b) there are factors which tend to make the effects from the model ore body larger than that of the actual ore bodies examined: estimated uranium reserves (factor of 3, but possibly more or less); and c) there are factors which do not affect the actual and model ore body estimates significantly: thickness and porosity of ore strata (each less than a factor of 2).

One obvious source of bias is the fact that the actual ore bodies discussed are of the relatively large size that make uranium milling economically feasible. A recent study by the Department of Energy (1979) indicates that small ore bodies are far more numerous than larger ones. Because information concerning the reserves, mineralization, location, etc., of ore bodies is generally considered proprietary, it is possible that consideration of only large ore bodies for this analysis has introducted other, unknown biases. Despite this, it is likely that the groundwater impact of the uranium ores, from which a 100,000-metric-ton heavy metal waste repository was derived, lies somewhere between the two estimates.

Conclusion

A simplified hydrologic model along with environmental transport and dosimetric models has been used to assess the environmental

impact of several actual uranium ore bodies. Although the impacts are only approximate, it is likely they represent the upper impact of ore bodies through groundwater pathways. The relatively large impact can be explained by the existence of ore bodies in aquifers. If ore bodies were not formed in aquifers, the effects of unmined ore would be both more difficult to access and much lower. Using this model the estimated fatal cancers vary for three actual ore bodies from a low of .59 to 3.4 fatal cancers/year; these fatal cancers result from estimated releases ranging from 190 millicuries to 1100 millicuries of ^{226}Ra per year.

A model ore body model would release 120 microcuries of ^{226}Ra per year. This would be the expected minimum impact from an ore body.

When normalized to the expected health effects of the ores from which a 100,000 metric-ton-heavy metal repository was derived, the number of fatal cancers ranges from 2.3×10^{-2} per year for the model case to 10^2 to 10^3 per year for actual ore bodies. It appears likely that the actual impact of individual ore bodies lies within this range, due to the assumptions made in the analysis.

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